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Contrasting land use legacy effects on forest landscape dynamics in the Italian Alps and the Apennines

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Abstract

Context: Land use legacies of human activities and recent post-abandonment forest expansion have extensively modified numerous forest landscapes throughout the European mountain ranges. Drivers of forest expansion and the effects of changes on ecosystem services are currently debated.

Objectives: i) to compare landscape transition patterns of the Alps and the Apennines (Italy), ii) to quantify the dominant landscape transitions, and iii) to measure the influence of climatic, topographic and anthropogenic driving factors.

Methods: Land cover changes and landscape pattern modifications were investigated at the regional (over 28 years, Alps and Apennines, Corine Land Cover dataset) and landscape scale (over 58 years, 8 Alpine and 8 Apennine sites, aerial images). The main driving factors of post-abandonment forest landscape dynamics were assessed with a statistical modeling approach.

Results: Forest expansion was the dominant landscape transition at both Italian mountain ranges, with an annual overall rate of 0.6%. Forest expansion was more extensive at lower elevations in the Apennines where climate is less limiting and extensive abandoned croplands and pastures were available throughout the study period. Distance from pre-existing forest edges in the Alps and elevation in the Apennines emerged as the most important predictors.

Conclusions: Forest expansion is most rapid where areas of recent agricultural abandonment coincide with favorable climatic conditions. Thus the prediction of forest landscape dynamics, in these mountain forests with a long history of cultural use, requires knowledge of how the magnitude and timing of land use changes intersect spatially and temporally with suitable conditions for tree establishment and growth.

Keywords Forest expansion, cultural landscape, historical ecology, aerial photographs, landscape structure, land abandonment

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Introduction

Land use history exerts a strong long term legacy on forest landscapes, affecting their structure, spatial pattern and associated ecosystem services (Bellemare et al. 2002; Ziter et al. 2017). At a landscape scale, forests after their removal can be replaced by other land cover types such as crops, pastures, and settlements, which in turn can lead to habitat fragmentation. Land abandonment has important effects on natural disturbance regimes especially in regions with a long history of intensive human influence (Mantero et al. submitted). After land abandonment, secondary succession can lead to forest expansion, a complex dynamic process influenced by different factors such as climate, topography, seed availability and anthropogenic variables. An important driver at local scales has proven to be the former land use intensity and type (Walker et al. 2010). For example, abandoned cropland, with compacted soil due to former plowing, may experience slower successional dynamics than former pastures (Dupouey et al. 2002). A pan-European scale study over the last 25 years revealed that the most important landscape transitions are urbanization and natural afforestation processes, both affecting landscape service provision (Van der Sluis et al. 2019). In forest landscapes post-abandonment processes generally cause increased wildfire frequency, extent and severity (e.g. Moreira 2001; Lloret 2002; Pausas et al. 2012) and a decrease in the frequency and intensity of rockfall (Lopez-Saez et al. 2016; Farvaque et al. 2019) and avalanches (Kulakowski et al. 2011; García-Hernández et al. 2017). Post-abandonment forest expansion can also cause changes to major bio-geochemical cycles, soil properties and eco-hydrological processes (Pellis et al. 2019), loss of biodiversity especially in semi-open areas with species-rich grasslands, and loss of cultural landscapes (Otero et al. 2015; Hermoso et al. 2018; Ridding et al. in press). Secondary succession processes can result in multiple pathways and abandoned lands may become more vulnerable to invasive species and fire (Munroe et al. 2013).

For thousands of years, anthropogenic pressure over the Mediterranean basin has shaped the numerous and diverse cultural landscapes (Naveh 1995). In many European rural areas abandonment after WW2 was a widespread socio-economic process, causing large human migrations toward urban areas (MacDonald et al. 2000; Poyatos 2003; Hochtl et al. 2005). The decline of traditional agro-pastoral activities was particularly intense in southern European mountains such as the Italian Alps and the Apennines. These two mountain ranges, covering approximately 35% of the entire country of Italy (Vacchiano et al. 2017), experienced a significant forest expansion after an extensive decrease of cultivated lands due to depopulation (Falcucci et al. 2007). Forest cover in Italy shifted from 6 million ha in 1936 (Forest Map of the Italian Kingdom) to 8.5 million ha in 1985 (First Italian Forest Inventory, IFNI85), and to 10.5 million in 2005 (Second Italian Forest Inventory, INFC05) and is currently estimated at about 11 million ha in 2015 with an increase of 20% in the last 30 years (Ferretti et al. 2018). These estimates rely on a wide range of sources, and studies using consistent datasets to quantify changes in land cover across broad areas prior to widespread availability of satellite imagery are lacking. In the Italian and French Alps, the depopulation of marginal lands started after the Industrial Revolution in approximately 1871 and, due to the two World Wars, lasted through the 1950s (Batzing et al. 1996). Before the 1950s, grazing was widely distributed in the Italian Alps where rangelands occupied 53% of the mountain areas (White 1950; Garbarino et al. 2013). This was primarily cattle grazing in unfenced pastures. In the Apennines a human migration from mountain areas toward the Adriatic and Tyrrhenian coastal areas occurred from 1951 to 1991, especially in the northern and southern Apennines (Malandra et al. 2018; Vitali et al. 2018). Many of these areas were subjected to former heavy exploitation for firewood and charcoal production, with wood pastures occurring frequently at higher elevations. A national reforestation program to reduce slope erosion, launched before WW2 and lasting until the 1970s, resulted in approximately 760,000 ha of new plantation forests composed mainly of coniferous tree species (Piermattei et al. 2016).

In both mountain regions, forest expansion has occurred as a gap-filling process at lower elevations and as an upward shift of treeline at higher elevations (Tasser et al. 2005). Forest expansion caused a direct reduction of open areas, reduced the extent of forest-grassland ecotones, and led to decreases in species diversity as well as culturally important landscapes (Falcucci et al. 2007; Petanidou et al. 2008). However, there are fundamental differences in how forest expansion processes have unfolded in the two mountain regions. A gentler topography and a favorable climate led to more intense deforestation in the Apennines, creating open areas that were used as pastures and crops both at low and high elevations. However, landscape mosaic simplification due to forest gap-filling processes mainly occurred at lower elevations. At higher elevations, forest succession on abandoned croplands and grasslands led to a complex and fragmented landscape (Malandra et al. 2019; Vitali et al. 2019). Understanding the underlying drivers of forest landscape changes by comparing the different land use legacies of the Alps and the Apennines is a fundamental step for more ecologically based landscape planning and management oriented toward biodiversity conservation and other ecosystem services.

The Alps and the Apennines are large and highly representative areas for testing our three hypotheses on mountain forest landscape changes in Italy over the last 60 years: 1) forest cover is increasing everywhere, but with different patterns in the Alps and the Apennines; 2) pasture-to-forest is the dominant landscape transition at high elevation; 3) historical forest cover (i.e. land use legacy) is the most important environmental driver for predicting forest expansion today.

Methods

Study area and sampling design

Our multiscale research design was structured at two spatial scales (region and landscape) aimed at comparing the two mountain ranges of Italy, the Alps (AL) and the Apennines (AP). They have similar total length (1300 – 1350 km, respectively), but different geographic orientation (AL: from west-to-east across northern Italy; AP: northwest-to-southeast from Liguria to Calabria). The two mountain ranges differ in terms of climate, topography and land use history.

In AL, mean annual temperatures range from less than 0° to over 10° C, with very cold winters. Annual precipitation ranges from 400 to over 3000 mm and summer dry periods are very rare (Isotta et al. 2013). Metamorphic lithology with intrusive igneous rocks prevail in the inner sectors and sedimentary outcrops dominate in the outer ones. Oak forests dominate at lower elevations whereas beech-silver fir (*Fagus sylvatica* and *Abies alba)* forests prevail on mesic aspects of the montane zone, replaced by *Pinus sylvestris* on xeric slopes. Coniferous forests with *Picea abies* (L.) H.Karst., *Larix decidua* Mill. and *Pinus cembra* L. dominate the subalpine zone (Fauquette et al. 2018).

At AP, mean annual temperatures range from 6° to 10° C, and annual precipitation ranges from 730 to 877 mm, with a short and pronounced summer dry period at lower elevations (Blasi et al. 2014). The eastern Mediterranean side (Adriatic) is generally more continental and humid than the western (Tyrrhenian) one. The forest vegetation is largely dominated by broadleaf species of the Mediterranean and temperate biomes. Xeric oak forests of *Quercus ilex* L., *Quercus pubescens* Willd., *Quercus cerris* L. and *Ostrya carpinifolia* Scop. dominate at lower elevations and *Castanea sativa* Mill. the sub-montane zone. *Fagus sylvatica*, locally mixed with *Abies alba*, largely dominates the montane zone up to treeline except for a few locations in the central and southern sectors where natural pine forests occur, dominated by *Pinus mugo*, *P. heldreichii*, or *P. nigra laricio*.

For our regional scale analyses, the study area in each mountain chain included all contiguous land above 500 m a.s.l. excluding those island polygons separated from the main mountain chains (Fig. 1). We obtained two large areas of 52,002 km^2 (AL) and 44,615 km^2 (AP), where we assessed the land-cover changes (LCC) for the 1990 – 2018 period, based on the Corine Land Cover (CLC Level 3, Feranec et al. 2016) raster maps

(100-m spatial resolution) after merging the original CLC categories into five larger groups: forest (FO), grassland (GR), cropland (CR), urban (UR), and unvegetated (UV) (Table S1). We developed a transition matrix for both regions by assessing changes for the five selected land-cover categories, allowing us to compute the relative changes in AL and AP.

For our landscape-scale analyses, in each region (AL and AP) we selected 8 landscapes of variable extent ranging from 6.3 to 16 km^2. These landscapes were selected and harmonized from previous projects and unpublished data on land-use/land-cover changes in AL and AP (e.g. Garbarino et al. 2013; Malandra et al. 2019. The 16 study landscapes cover a total surface area of approximately 23,000 ha within an elevation range of 500 – 2,600 m a.s.l., including all vegetation zones (Table 1). We adopted the altitudinal threshold of 2,600 m a.s.l., as the potential alpine treeline (Caccianiga et al. 2008; Lingua et al. 2008; Garbarino et al. 2013) in order to limit the LUC analysis to the vegetated part of the 16 landscapes and to reduce the weight of the 'rock' land cover category, which is uninformative for our research. Historical aerial photographs for the years 1954-1962 (b/w, 1: 60,000 approximate cartographic scale, Italian Geographic Military Institute - IGMI) were scanned and digitized at 800 ppi, with a mean spatial resolution of 1 m. The IGMI images were georeferenced and orthorectified using PCI Geomatica 2012 software. Regularly distributed tie points were used to co-register IGMI images with 2012 orthophotos (RGB, 0.5 m cell size, National Agency for Funding in Agriculture - AGEA) resampled at 1 m. The average horizontal Root Mean Square Error (RMSE) was 23 m± 2SD. We used the TINITALY DEM at 10 m spatial resolution (Tarquini et al. 2012) for orthorectification.

We applied a semi-automated object-based classification by combining the automated image segmentation from eCognition software (scale factor 100, color factor 0.5) with on-screen photointerpretation of segmented polygons (Garbarino et al. 2013). We then performed a supervised classification of the objects based on an initial set of at least 10 training polygons for each category, selected through photointerpretation. This was followed by a manual classification of the previously unclassified polygons. Each polygon of the 32 land cover maps (16 landscapes × two time periods), was classified into five land cover classes that were used for the regional-scale analysis: forest (FO), grassland (GR), cropland (CR), urban (UR), unvegetated (UV). The UV category includes different land cover types such as rock, gravel, sand, bare soil and sparse vegetation areas. The latter is a mosaic of sparse grasslands and barren nonvegetated areas that are mostly located between 2000-3000 m a.s.l. A post-processing procedure on the resulting 32 land cover maps was performed in a GIS environment to enforce consistency among the two datasets. A minimum mapping unit (MMU) of 100 m^2 (Garbarino et al. 2011) was obtained by merging smaller polygons with neighboring larger ones by using the ArcGIS tool 'Eliminate' (Malandra et al. 2019). Merged polygons were rasterized at 1-m resolution and the resulting raster maps were smoothed using a moving window (3 × 3) majority filter. We obtained the level of accuracy by randomizing 16 polygons/ha on each map and classifying the objects visually using the same land cover categories adopted in the automatic segmentation (Radoux and Bogaert 2017). Overall, the classification accuracy ranged from 78% to 96% with a Cohen's Kappa coefficient between 0.67 and 0.93 (Table S2). The land cover change analysis at the landscape scale provided 16 transition matrices that were combined to detect overall transitions and differences between the AL and AP mountain regions. We converted the two transition matrices into two transition diagrams showing gain, loss, net change and persistence for each land cover category (Cousins 2001). With the same dataset (Garbarino et al. 2019), we computed the relative contribution (in hectares) of each land cover category to the transition to forest cover and we applied a Mann–Whitney test to compare the medians of the two mountain ranges (AL and AP). The Mann–Whitney test was performed for each of the five categories by using the 8 landscapes as sample size for each mountain range.

For our landscape-scale analyses, we assessed the main drivers affecting forest expansion in AL and AP using a Random Forest (RF) model (Rodman et al. 2019). Specifically, we modelled the occurrence of a transition to forest through a binary classification method using 'mlr' (Bischl et al. 2016) and 'ranger' (Wright & Ziegler 2017) R packages (R Core Team 2019). Given the unbalanced ratio between cells with a transition to forest (minority class) and cells that remained unchanged (the majority class), we under-sampled unchanged cells using a spatially random selection within each landscape using the ‘spatialEco’ R package (Evans 2019). Before computing class transitions, we filtered out from the dataset the landscape portions that were already forests in the past, and we downscaled gridded land cover maps from 1 m to 30 and 60 m resolutions using the majority class within each coarser cell. The coarser resolutions allowed us to limit the influence of both co-registration and classification errors of aerial images on model predictions and to evaluate the dependence of model predictions on the spatial scale. For our final analysis, we used the 30 m resolution because RF models trained with data at 30 or 60 m resolutions produced very similar outputs in terms of variables importance and trends of partial dependences. At a coarser resolution prediction errors slightly increased (Table S3).

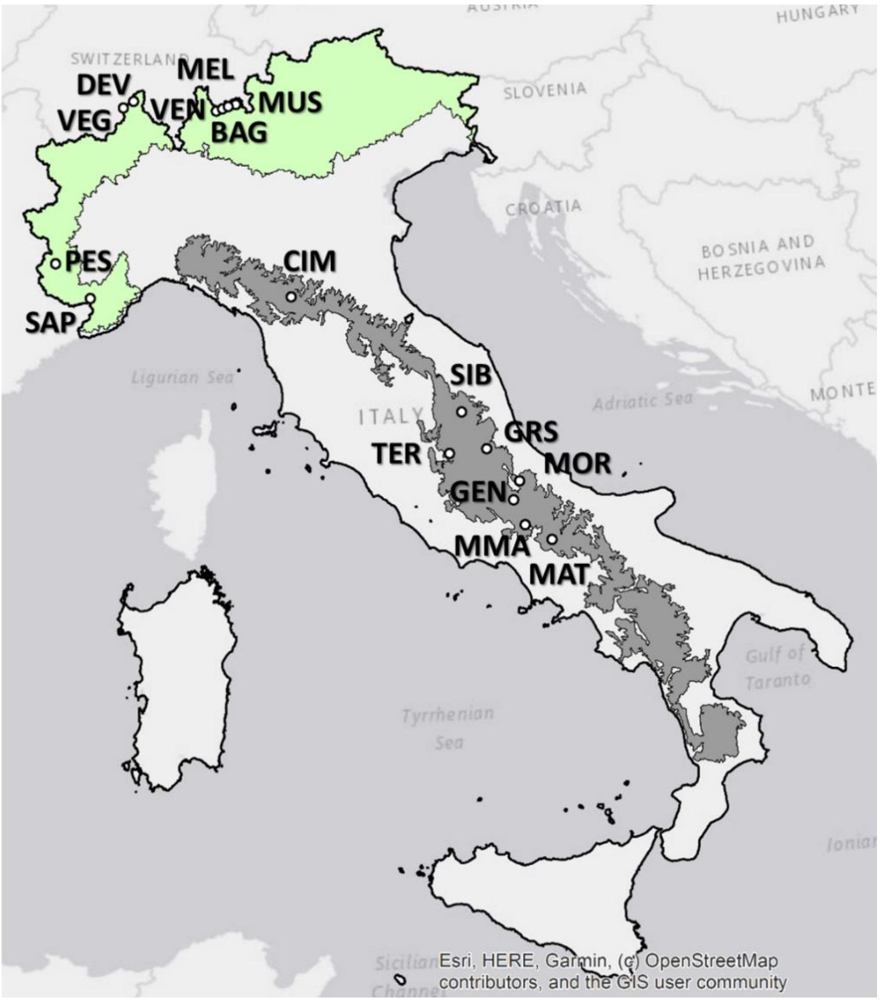

Figure 1. Location of the 16 landscapes (white circles) within the two Italian mountain regions: the Alps (light green) and the Apennines (light grey) with a minimum elevation of 500 m a.s.l. For landscape codes see Table 1.

Because there was a substantial area of *Pinus* plantations within the forest cover of 2012 images in certain landscapes of the Apennines, we removed plantation patches before computing class transitions in order to model only natural forest dynamics.

Table 1. Environmental descriptors (Area = total surface area; El = mean elevation; Sl = mean slope; Te = mean annual temperature; Pr = mean annual precipitations; BD = mean distance from buildings; RD = mean distance from roads) of 16 landscapes divided by mountain region (AL or AP).

Region	Landscape name	Landscape Code	Area (ha)	El (m a.s.l.)	Sl (°)	Te (°C)	Pr (mm)	BD (m)	RD (m)
AL	Bagni	BAN	1574.3	1788.1	36.0	3.9	1081.3	678.4	289.4
AL	Mello	MEL	1433.7	2045.8	34.1	1.7	983.7	935.0	359.8
AL	Sapè	SAP	959.4	1419.8	22.3	6.2	1001.9	343.8	178.4
AL	Pesio	PES	1599.2	1596.5	31.7	5.7	980.0	711.0	202.1
AL	Ventina	VEN	630.4	2248.6	30.1	0.1	900.8	1044.9	366.4
AL	Musella	MUS	921.4	2155.6	27.7	2.0	788.2	592.1	190.6
AL	Veglia	VEG	1407.2	2043.8	25.4	1.7	1045.1	815.3	205.1
AL	Devero	DEV	1570.0	2169.9	23.6	1.2	1138.6	1048.2	323.3
AP	Cimone	CIM	1593.4	1444.7	19.2	6.8	1352.5	349.3	111.6
AP	Sibillini	SIB	1601.3	1397.2	26.3	6.8	940.1	549.4	115.9
AP	Gran Sasso	GRS	1602.4	1577.5	27.2	6.3	872.3	1548.4	294.2
AP	Terminillo	TER	1600.7	1573.5	25.7	6.8	831.5	700.4	202.8
AP	Morrone	MOR	1603.7	1422.8	22.8	7.6	792.4	1447.3	276.9
AP	Genzana	GEN	1603.4	1234.7	22.4	8.2	789.1	664.7	228.5
AP	Monte Mare	MMA	1604.0	1149.7	24.1	9.0	780.9	1622.2	410.9
AP	Matese	MAT	1605.4	1112.5	22.5	9.2	694.7	481.4	163.4

We used several spatial predictors (Table S4) such as the distance from pre-existing forest edges, topographic variables (elevation, slope, heat load index *sensu* McCune and Grace 2002), climatic variables (mean annual temperature, annual precipitation), and anthropogenic variables (cost of movement, Euclidean distance to buildings, Euclidean distance to roads). We derived topographic variables from the 10 m DEM and climatic variables from the 'Climatologies at high resolution for the earth's land surface areas' (CHELSA) v1.2 datasets at 30 arcsec (~1 km) spatial resolution (Karger et al. 2017). We computed the accumulated cost of movement across the terrain through the Tobler's hiking function (on-path) implemented in the 'movecost' R package (Alberti 2019), using "buildings" in OpenStreetMap as starting locations. We applied two different approaches to obtain either the RF models predictions or the predictive performance estimates with a reduced bias. Specifically, we trained two RF models, one for AL and one for AP, using all the data and tuning hyper-parameters through an 8-fold spatial cross-validation (Brenning 2012, Schratz et al. 2019). We used bias-reduced predictive performance estimates using two common measures in binary classification, the Brier score (Brier 1950) and the area under the receiver operating characteristics curve (AUC). These measures were averaged over a total of 800 RF models obtained through an 8-fold spatial cross-validation repeated 100 times using a nested 5-fold spatial cross-validation for hyper-parameters tuning (Lovelace et al. 2019, Schratz et al. 2019). For both strategies, we used a sequential model-based optimization approach in 'mlrMBO' R package (Bischl et al. 2017) to search for the optimal RF hyper-parameters (mtry, sample fraction and minimum node size) using 50 steps. The spatial cross-validation resampling technique was based on k-means clustering of observation coordinates and allowed us to geographically partition the data, thus maintaining the assumption of independence among training and test sets which would be violated in the case of randomly sampled observations due to the presence of spatial autocorrelation. We assessed variable importance from each model using the permutation method (Breiman 2001) and we employed partial dependencies (Friedman 2001; Goldstein et al. 2015) to interpret the marginal effect of each variable on the predicted probability of forest expansion. Specifically, we computed the average and the standard deviation of individual marginal effects obtained using all the observations in the dataset through the 'generatePartialDependenceData' function in 'mlr' R package.

Results

Regional land cover changes – CORINE database

Forest expansion occurred in both areas, but was greater in AL (+2,951 ha, +9 %) than in AP (+1023 ha, +3.7 %) (Tab. 2). Cropland cover is generally stable whereas human infrastructures increased more at AP (+ 131 ha, +34 %) than at AL (+256 ha, +19 %). Grasslands greatly decreased at both mountain ranges, but more at AL (-2784 ha, -39 %) than at AP (-1160 ha, -27 %), a pattern also observed for unvegetated areas (AP -56 ha, -19 %; AL -503ha, -9 %). Regional scale results (1990-2018) were weakly in agreement with the landscape scale results (1954-2012) as shown in the supplementary material (Table S5).

Table 2. Land cover categories surface areas expressed as a percentage of the total mountain area derived from the Corine Land Cover in the Alps (AL) and the Apennines (AP) throughout the years. Changes across the entire period (1990-2018) are indicated in the last two columns as absolute and relative percent values.

AL	1990	2000	2006	2012	2018	1990-2018 Change	
						Absolute	*Relative*
Forest (FO)	62.9	62.4	62.5	68.6	68.6	+5.7	+9.0
Grassland (GR)	13.8	14.9	14.6	8.5	8.5	-5.4	-38.8
Cropland (CR)	10.1	10.1	10.4	10.2	10.2	+0.2	+1.5
Urban (UR)	2.6	2.7	2.8	3.0	3.1	+0.5	+19.3
Unvegetated (UV)	10.6	9.9	9.7	9.6	9.6	-1.0	-9.2
AP	1990	2000	2006	2012	2018	1990-2018 Change	
						Absolute	*Relative*
Forest (FO)	62.5	62.5	62.3	64.8	64.8	+2.3	+3.7
Grassland (GR)	9.8	9.8	9.8	7.2	7.2	-2.6	-26.6
Cropland (CR)	26.2	26.1	26.2	26.3	26.3	+0.1	+0.5
Urban (UR)	0.9	1.0	1.1	1.1	1.2	+0.3	+34.2
Unvegetated (UV)	0.7	0.6	0.5	0.5	0.5	-0.1	-19.1

Landscape transitions – aerial imagery

Forest expansion also occurred at the landscape scale in both ranges. For all study landscapes combined, the mean annual forest surface area increment was 60 ha/year (0.5 %/year), and was slightly greater at AP (41 ha yr^{-1} or 0.6% yr^{-1}) than at AL (19 ha yr^{-1} or 0.4% yr^{-1}). Grasslands and Crops decreased in both areas, but a larger reduction of crops (CR) occurred at AP (Fig. 2). Unvegetated lands (UV) decreased only at AL, and urban infrastructures (UR) increased more at AP. The relative weights of CR and UR were historically higher at AP, whereas UV values were historically higher at AL. Forest expansion was mostly related to the "GR to FO" transition (66.3 % overall), but was greater at AL than at AP (74.4 % and 62.4 % respectively, Fig. 3). The "CR to FO" transition was stronger at AP than AL (32.4 % and 1.4 %), whereas the opposite pattern was observed for the "UV to FO" transition (AL = 23.9 %, AP = 3.1 %). All land cover transitions to forest were significantly different between the two mountain ranges (Mann-Whitney test: GR, UR with $p < 0.05$; CR, UR with $p < 0.01$).

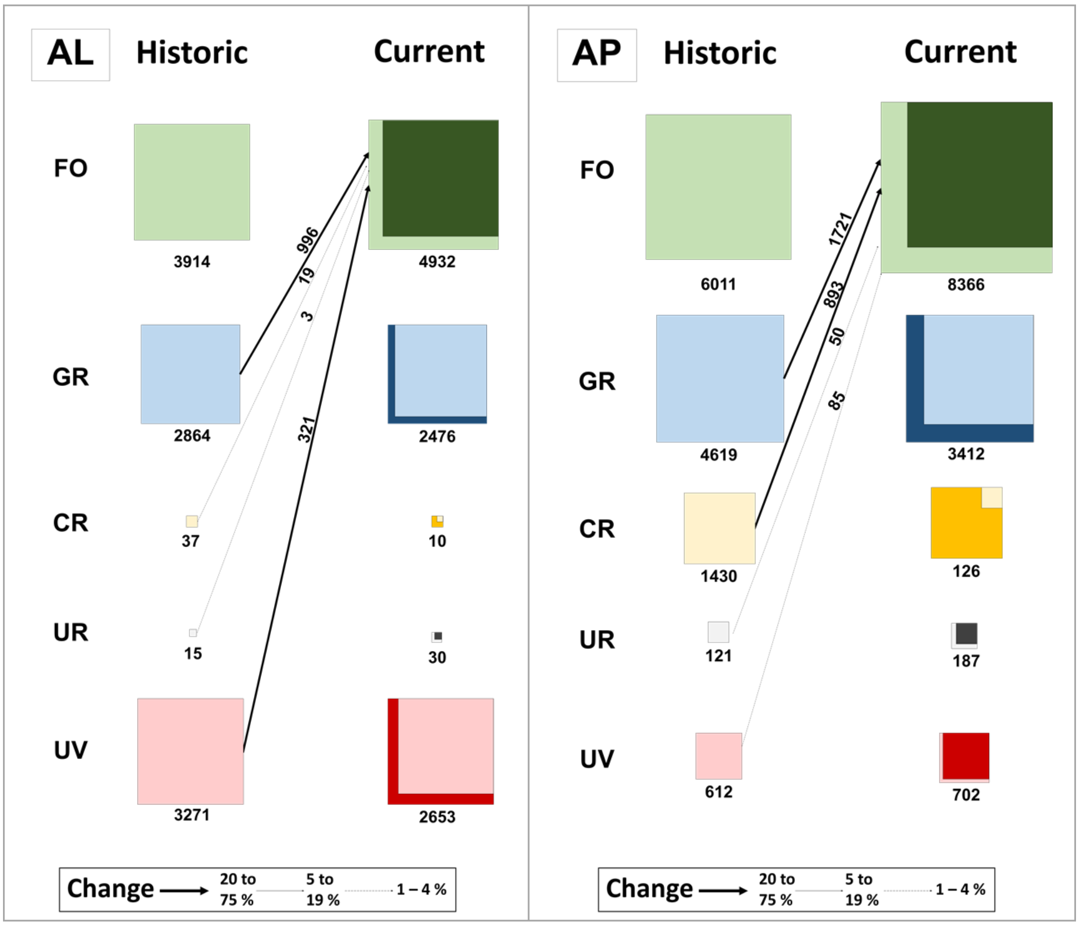

Figure 2. Area of land cover classes (ha) and land cover transitions from historic (1954-1962) to present time (2012) in the Alps (left panel) and the Apennines (right panel) study sites. Colored boxes refer to land cover categories with box size scaled to area: darker-colored inset boxes represent land cover class (LCC) persistence over time in the case of LCC increase (e.g. FO and UR categories); light-colored inset boxes represent persistence over time in the case of LCC decrease (e.g. GR, CR and UV). Transitions representing forest expansion are highlighted with arrows. Arrow thickness increases with magnitude of the land cover transition. The area converted to forest (ha) for each transition is reported in text above the arrow symbols.

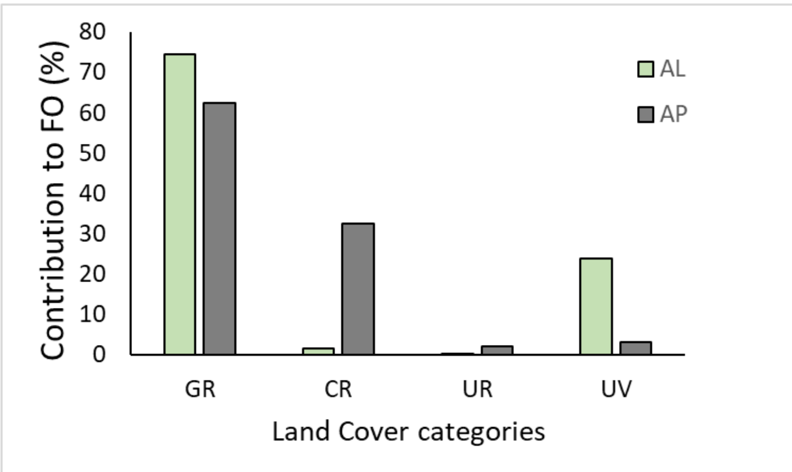

Figure 3. Percent contribution of each land cover category (GR = grassland, CR = cropland, UR = urban, UV = unvegetated) to forest expansion over the studied period (1954-2012).

Drivers of forest expansion

RF models derived for the forest expansion portion of the landscape indicate that the best predictor was the distance from pre-existing forest edges (DF), particularly at AL (importance rate - *IR* = 0.13, Fig. 4). Forest expansion is predicted also by elevation (El), which is particularly important for AP mountain range (AL: *IR* = 0.08, AP: *IR* = 0.16). Climatic variables such as mean annual temperature (Te) and precipitation (Pr) were more important at AP (*IR* = 0.07-0.08). Anthropogenic impact proxy variables (MC, DB and DR) were less influential in the models (*IR* = 0.05-0.03), but DB was more important for AP than for AL (*IR* = 0.05 and 0.03 respectively).

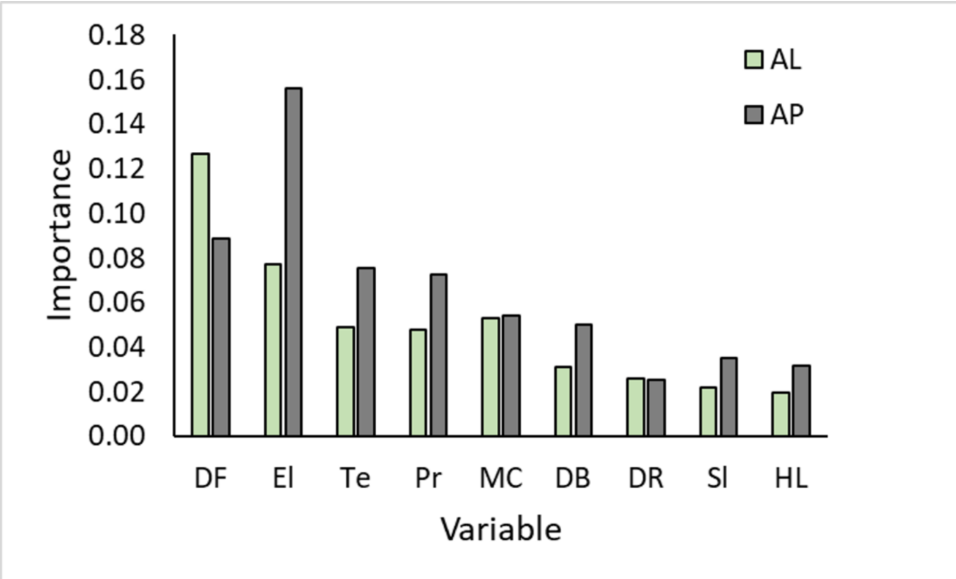

Figure 4. Importance rate of variables in random forest (RF) models of the Alps (light green) and the Apennines (grey). Variables are: distance from pre-existing forest edges (DF), topographic (El = elevation, Sl = slope, HL = Heat load index), climatic (Te = mean annual temperature, Pr = annual precipitation) and anthropogenic (MC = cost of movement, DB = Euclidean distance to buildings, DR = Euclidean distance to roads).

At AL forest expansion probability was higher close to pre-existing forest edges, rapidly declining between 0 and 200 m and with a gradual decline between 200 and 800 m (Fig. 5). At AP the negative relationship between forest expansion probability and distance to pre-existing forest edges featured a rapid decline (from 0.6 to 0.4 of probability) between 0 and 150 m and a gradual decline between 150 and 900 m. The effect of distance from pre-existing forest edges (DF) observed at AP exhibited a higher heterogeneity compared to those observed at AL as highlighted by standard deviation computed at different values of the predictor variable (Fig. 5). Forest expansion at AP was more likely to occur at lower elevations (500 – 1,000 m a.s.l.) and the probability abruptly decreased between 1,000 and 1,500 m a.s.l. A similar pattern was observed at AL, but at higher elevations (1,000 – 2,000 m a.s.l.), with a clear decrease from 2,000-2,500 m a.s.l. Relationships between forest expansion probability and annual temperature were generally weak, although there was a slightly higher probability of forest expansion occurring between 2.5 - 7 °C at AL and 7.5 - 13 °C at AP.

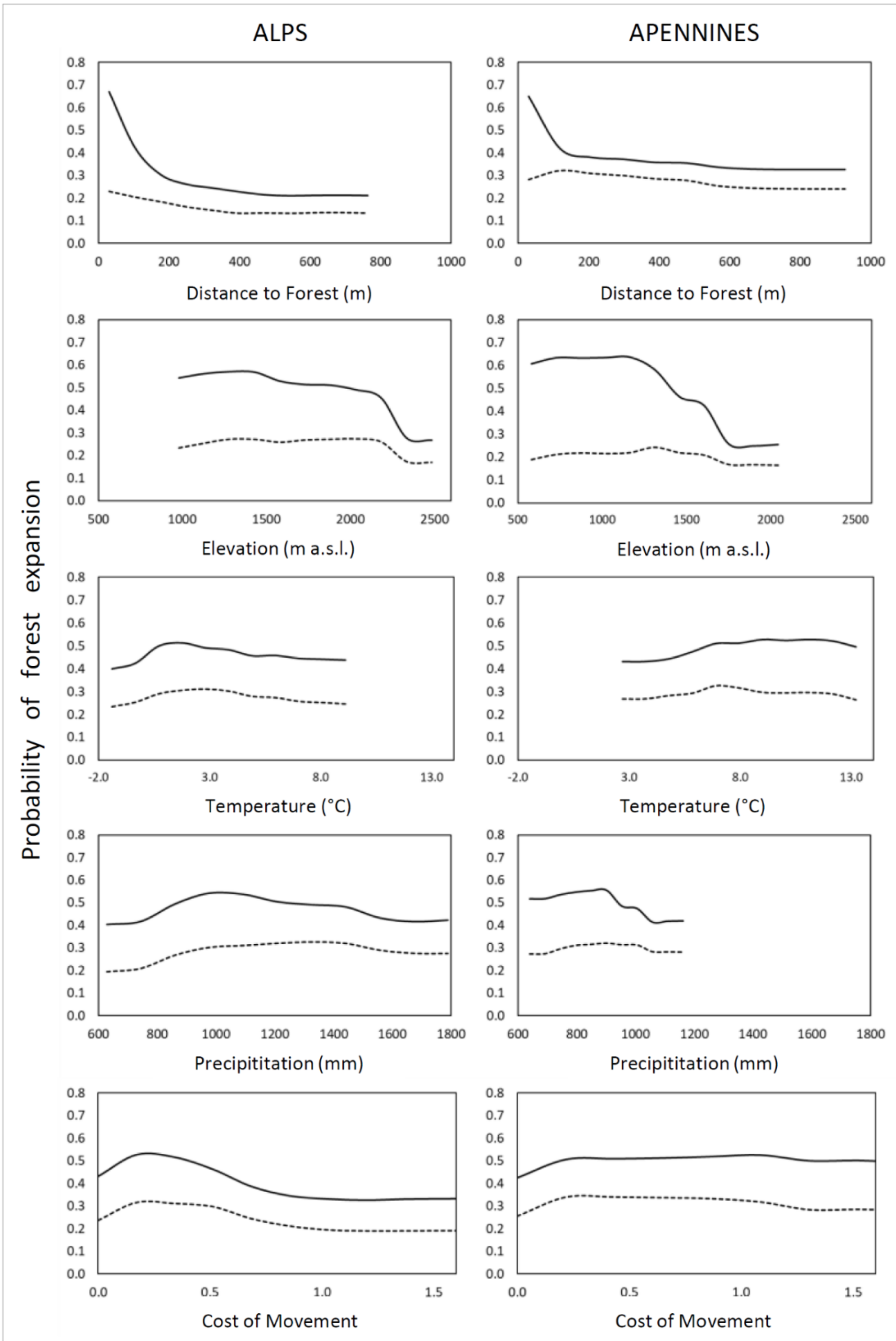

Figure 5. Partial dependence plots showing relative influences of the five most important predictors on the probability of forest expansion, for the Alps (left) and the Apennines (right) across the respective input data ranges. We used all observations to train the model for computing the partial dependence function. Solid lines indicate the average over individual marginal effects of each selected explanatory variable (distance from pre-existing forest edges, elevation, temperature, precipitation and cost of movement) whereas dotted lines are the standard deviations of individual marginal effects.

Discussion

Forest expansion following land abandonment is a well-known process in mountain forests globally (Sitzia et al. 2010). For example, forest expansion in mountainous populated areas has been recently detected in South America (Nanni et al. 2019) and East Asia (Fang et al. 2014). Grazing decline and fire suppression favored forest expansion and forest infilling in California, USA (Lydersen and Collins 2018) and recently rain forest expansion into savanna has been detected in Australia (Ondei et al. 2017). Natural forest expansion is particularly evident and rapid in several mountain ranges of the World such as in the Mediterranean Basin (e.g. Roura-Pascual et al. 2005; Niedrist et al. 2009; Weisberg et al. 2013) where agro-silvo-pastoral traditional practices declined abruptly due to rural depopulation (Lasanta et al. 2017). However, there have been few studies comparing post-abandonment forest expansion patterns among different regions or mountain ranges (e.g. Tasser et al. 2007; Fontana et al. 2014).

By comparing the Italian Alps and the Apennines we found that environmental influences on forest expansion processes were similar between the two regions. Our results for the observed time span (1954-2012) indicate an overall forest area increase of 0.6% yr^{-1} in the Italian Alps and Apennines. These values match with the annual increments recently reported either for the entire Italian peninsula in 1985-2005 (0.3% yr^{-1}) and 2005-2015 (0.2% yr^{-1}) periods (RAF 2018) or for different sites of the Apennines (0.4 - 0.7% yr^{-1}, Brachetti et al. 2012; Malandra et al. 2019). Similar rates are reported for other Alpine regions such as Carnia (0.7% yr^{-1}), Tyrol (0.35% yr^{-1}) and South Tyrol (0.1% yr^{-1}) in the 1955-2000 period (Tasser et al. 2007).

Important differences between the two studied mountain ranges emerged from our analysis. Forest expansion was more intense in the Apennines during the 1954-2012 period. This outcome could be related to the different geographic layout of this mountain range and the wider latitudinal gradient that it encompasses, providing warmer climate conditions more favorable for forest regeneration. Another possible explanation for this pattern arises from to the large differences in elevation gradients between the two mountain ranges. Abandoned sites at lower elevations that were previously cultivated or grazed were more common in the Apennines than in the Alps, and such sites experienced more rapid and extensive forest expansion.

These differences highlight the importance of regional variation in climate and land use history for understanding and predicting forest landscape change following agricultural abandonment. Differences in land use history expressed by a mosaic of former croplands and pastures have important long-term implications for post-abandonment forest establishment (Zimmermann et al. 2010). At the regional scale, we found a greater reduction of grasslands in the Alps than in the Apennines, where we found a greater increase of anthropogenic land cover types (mostly UR). The landscape transitions from grasslands, croplands, and unvegetated lands to forests were by far the most relevant at our landscape scale of analysis. Grassland-to-forest was the dominant shift in both mountain ranges due to a general decline of traditional cattle grazing in mountain areas (e.g. Nagy et al. 2003). However, the two mountain ranges differ in that the widespread transition from unvegetated areas (e.g. rocks and bare soil) to forest occurred only in the Alps. Here, this transition is probably due to the tendency for coniferous treeline species (*Larix decidua* and *Pinus cembra*) to invade higher-elevation, shrub-dominated and alpine plant community types (Vittoz et al. 2008). On the other hand, *Fagus sylvatica* dominated treelines of the Apennines are less prone to upward migration; forest expansion here was mostly the outcome of gap infilling processes (Vitali et al. 2018, Malandra et al. 2019). High elevation forests in the Apennines are dominated by *Fagus sylvatica*, a strongly resprouting species, but with heavy seeds that disperse predominantly over short distances (Vitali et al. 2017). Conifer species with greater long-distance seed dispersal ability occur at a few sites of the central (*Pinus nigra* – Vitali et al. 2017; *Pinus mugo* – Dai et al. 2017) and southern (*Pinus heldreichii* – Vitali et al. 2018) Apennines. Here, croplands-to-forest was the second most important transition, given the possibility of growing a few rare food crops (e.g. potatoes, special cultivars of cereals, apples and chestnuts) at higher elevations especially on southern or less exposed slopes (Bracchetti et al. 2012; Rovelli 2019). The abandonment of upland traditional farming systems in the Apennines is one of the most important socio-economic drivers of landscape degradation and biodiversity depletion (Farina 1995; Zimmermann et al. 2010).

Pre-existing forest edges emerged as a key land use legacy for future forest expansion both in the Alps (Abadie et al. 2017; Tasser et al. 2007) and the Apennines (Malandra et al. 2019) with a greater importance in the Alps. A general explanation for this is related to seed source availability and to the marginality of ecotones such as forest edges. These are the first pastoral zones to be abandoned when grazing pressure is reduced. The distance from pre-existing forest edges (years 1954-1962) appears a stronger driver in the Alps where, because of harsh conditions at higher elevations, favorable microsites are necessary for tree establishment. In the Apennines, where this variable was second in importance, high variability in effect size is likely caused by heterogeneity among individual observations belonging to different landscapes. On the other hand, in the Apennines, elevation was the most influential predictor variable with widespread forest expansion having occurred at lower altitudes on slopes severely exploited prior to the analyzed time span (1954-2012). The probability of forest expansion gradually decreased along an altitude gradient from 1,000 to 1,800 m a.s.l. in the Apennines, but decreased abruptly between 2,000 and 2,300 m a.s.l. in the Alps. Forest expansion on former pastures and croplands was also faster at lower elevations in mountains of southern Spain (Fernàndez et al. 2004).

The importance of land use legacy for forest landscape dynamics is emphasized by long-term studies demonstrating that legacies may persist for decades, affecting current and future land cover changes (Loran et al. 2017; Tasser et al. 2017). Human activities such as harvesting, grazing, fire and litter removal when practiced for long time periods, may greatly affect current forest dynamics (Gimmi et al. 2013). The role of land use legacies on past and future forest dynamics is typical of many southern European mountain ranges shaped by historical anthropogenic disturbance regimes such as an intensive land use (Albert et al. 2008; Ameztegui et al. 2016). Our study confirms the importance of the location of pre-existing forest edges as the legacy of centuries of human land use in mountain regions, as in other Mediterranean mountain ranges such as the Pyrenees and the French Alps (Mouillot et al. 2005; Gartzia et al. 2016; Abadie et al. 2017).

The differences observed in land cover change patterns between the Alps and the Apennines are not surprising because of strong regional differences in climate, geology, topography, vegetation (e.g. treeline species composition). Forest expansion by upward treeline rise or forest gap-filling processes has occurred primarily on warmer and gentler slopes (e.g. southern exposure), whether in the Alps (Tasser et al. 2007; Garbarino et al. 2013), the Apennines (Vitali et al. 2018) or the Pyrenees (Gartzia et al. 2016). More favorable climate conditions and the greater availability of abandoned open areas make south-exposed low elevation sites suitable areas for forest expansion. Ultimately, the more rapid rate of forest expansion in the Apennines was linked to the greater availability of open areas given the more intense previous land use. Tree encroachment on old pastures by secondary succession and on former unvegetated areas through primary succession prevail in the Alpine region where climate change appears to have a strong influence (Dirnböck et al. 2003; Gehrig-Fasel et al. 2007; Giorgi and Lionello 2008). Elevation plays a fundamental role on forest dynamics at both landscape and stand scales in the Alps (Garbarino et al. 2009; Kulakowski et al. 2011). Land use legacies such as conifer plantations and high elevation crop farming seem stronger in the Apennines where climate change effects on forest expansion at high elevation appear constrained by the unsuitability of *Fagus sylvatica* to migrate upwards due to its heavy seeds and its limited ability to invade adjacent plant community types (Vitali et al. 2018, 2019). However, the limited transitions from unvegetated areas to forests in the Apennines were also due to a combination of topographic influences and the previous land use. In particular, several Apennines peaks with sandy or marl-sandy soils are less topographically limited (mountain mass effect) so that unvegetated areas are rare and in the absence of past human activities they are climatically suitable for forest dominance.

Forest expansion is an emerging and debated issue that requires accurate measurement and monitoring to allow for proper management of current and future dynamics (Otero et al. 2015). There are two main management strategies: i) passive management to support rewilding processes and limit human induced landscape fragmentation and ii) active management to control and limit the negative effects of re-vegetation processes (Lasanta et al. 2015). Negative effects of natural forest expansion include simplification of landscape structure, decline of species diversity, increased risk of fire and soil erosion, and the loss of cultural landscapes (Lasanta et al. 2015; Ferretti et al. 2019). A balance between conservation through monitoring and active management of secondary succession dynamics (new forests) should be attempted. A recent review, contrasting active management strategies with passive strategies allowing forest secondary succession, found that the most efficient technique seems to be a combination of clearing and extensive grazing (Lasanta 2019), maintaining high levels of landscape complexity and forest-meadow edge.

With this study, by means of a standardized aerial image processing protocol we provided a robust dataset that should be implemented with more comprehensive records (Garbarino et al. 2019). Quantitative historical ecology with data on land use legacies can provide excellent information for ecosystem modelling to predict forest landscape changes (Stürck and Verburg 2017).

We have shown that forest expansion in mountain ranges of Italy is controlled by land use legacies of pre-existing forest edges, interacting with topography and climate. The Alps and the Apennines showed similar landscape changes featuring grassland-to-forest transitions. However, the rate of forest expansion was faster in the Apennines for the larger occurrence at lower elevations of old-fields recolonized by secondary forests. In the Alps, climate and land use changes favored a widespread transition from unvegetated areas to forest at higher elevations. Our results could be biased by the stronger mass effect on the Alps, and the higher average elevation of alpine landscapes. A further limit in our approach is the spatial resolution mismatch between the 1-km climate data resolution and the 30 m unit of forest expansion analysis. Thus, the influence of climatic variables on forest expansion could be underestimated by our random forest models.

Despite these limitations, our results demonstrate that post-abandonment forest expansion is a widespread and ongoing process in Italian mountain forest landscapes. Future research should increase the number of surveyed sites for increased sensitivity in comparing regional differences. It would be informative to apply a land use change modeling approach. Predicting new landscape scenarios for Italian mountain forests should account for the possible changes to disturbance regimes linked to climatic changes (Vacchiano et al. 2017). The extensive forest cover that is blanketing large mountain areas has important implications for habitat and biodiversity. Forest expansion in these mountain landscapes additionally leads to an increase of fuel load continuity that increases the risk of wildfires, particularly in areas exposed to severe drought stress increased by recent climate changes. Large wildfires have recently increased in occurrence in mountain areas where they have historically been quite rare due to the prevalence of managed pastures and farmlands. Other implications regard snowfall and accumulation regimes in combination with soil erosion dynamics after forest fire. These issues are connected also to the naturalness of these processes triggered by man in human-shaped landscapes. A no-management approach of the successional processes would not guarantee, at least in the short term, a strictly natural forest encroachment. A comprehensive overview and assessment of the multiple ecosystem services provided by these complex and millenary landscapes is necessary in order to attempt a sustainable forest management (Schulze and Schulze, 2010).

Author contribution

Matteo Garbarino conceived the experiment and all authors contributed to the study design. Material preparation, data collection and analysis were performed by Matteo Garbarino, Donato Morresi, Emanuele Sibona, Francesco Malandra, Alessandro Vitali and Carlo Urbinati. The first draft of the manuscript was written by Matteo Garbarino and all authors commented on previous versions of the manuscript. All authors read and approved the final manuscript.

Data availability

The datasets generated during and analyzed during the current study are available in the Figshare repository, https://doi.org/10.6084/m9.figshare.11409444.v1

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Supplemental material

Table S1. Harmonization details of the Corine Land Cover legend in 5 land cover categories at AL and AP.

Corine Land Cover		Harmonization	
Code	**Description**	**Code**	**Description**
111	Continuous urban fabric	UR	Urban
112	Discontinuous urban fabric	UR	Urban
121	Industrial or commercial units	UR	Urban
122	Road and rail networks and associated land	UR	Urban
123	Port areas	UR	Urban
124	Airports	UR	Urban
131	Mineral extraction sites	UR	Urban
132	Dump sites	UR	Urban
133	Construction sites	UR	Urban
141	Green urban areas	UR	Urban
142	Sport and leisure facilities	UR	Urban
211	Non-irrigated arable land	CR	Cropland
212	Permanently irrigated land	CR	Cropland
213	Rice fields	CR	Cropland
221	Vineyards	CR	Cropland
222	Fruit trees and berry plantations	CR	Cropland
223	Olive groves	CR	Cropland
231	Pastures	GR	Grassland
241	Annual crops associated with permanent crops	CR	Cropland
242	Complex cultivation patterns	CR	Cropland
243	Land principally occupied by agriculture with … natural vegetation	CR	Cropland
244	Agro-forestry areas	CR	Cropland
311	Broad-leaved forest	FO	Forest
312	Coniferous forest	FO	Forest
313	Mixed forest	FO	Forest
321	Natural grasslands	GR	Grassland
322	Moors and heathland	FO	Forest
323	Sclerophyllous vegetation	FO	Forest
324	Transitional woodland-shrub	FO	Forest
331	Beaches dunes sands	UV	Unvegetated
332	Bare rocks	UV	Unvegetated
333	Sparsely vegetated areas	UV	Unvegetated
334	Burnt areas	FO	Forest
335	Glaciers and perpetual snow	UV	Unvegetated
411	Inland marshes	UV	Unvegetated
412	Peat bogs	UV	Unvegetated
421	Salt marshes	UV	Unvegetated
422	Salines	UV	Unvegetated
423	Intertidal flats	UV	Unvegetated
511	Water courses	UV	Unvegetated
512	Water bodies	UV	Unvegetated
521	Coastal lagoons	UV	Unvegetated
522	Estuaries	UV	Unvegetated
523	Sea and ocean	UV	Unvegetated
999	NODATA	999	No Data

Table S2. Classification accuracy (OA = overall accuracy, K = Cohen's Kappa coefficient) of 32 land cover maps (16 landscapes x 2 periods).

Landscape	**Year**	***OA***	***K***
BAG	1962	79	0.69
	2012	89	0.83
CIM	1954	82	0.73
	2012	93	0.88
DEV	1954	82	0.71
	2012	86	0.77
GEN	1954	78	0.70
	2012	78	0.67
GRS	1954	80	0.67
	2012	86	0.76
MAT	1954	86	0.77
	2012	94	0.88
MEL	1962	82	0.73
	2012	88	0.82
MMA	1954	91	0.83
	2012	90	0.68
MOR	1954	82	0.69
	2012	90	0.83
MUS	1961	82	0.72
	2012	89	0.84
PES	1954	88	0.78
	2012	86	0.73
SAP	1954	84	0.73
	2012	91	0.84
SIB	1954	86	0.80
	2012	86	0.77
TER	1954	85	0.75
	2012	84	0.71
VEG	1954	87	0.81
	2012	91	0.87
VEN	1961	94	0.89
	2012	96	0.93

Table S3. Random forest model parameters used in this study (N transitions = number of forest expansion pixels, N total = number of total pixels in the model, OOB = Out-of-bag error or prediction error of RF models, Brier score = Brier predictive performance estimate, AUC = area under the receiver operating characteristics curve, K = Cohen's Kappa coefficient) computed for the AL and the AP datasets both at 30 and 60 m spatial resolutions.

RF models	AL 30 m	AP 30 m	AL 60 m	AP 60 m
N transitions	14420	21027	3555	5184
N total	28900	40575	7082	10010
OOB	0.108	0.091	0.126	0.098
Brier score	0.175	0.151	0.182	0.132
AUC	0.829	0.854	0.819	0.849
K	0.482	0.519	0.483	0.510
DF *IR*	0.127	0.088	0.114	0.072
El *IR*	0.077	0.156	0.065	0.168
MC *IR*	0.053	0.054	0.034	0.039
Te *IR*	0.049	0.076	0.033	0.051
Pr *IR*	0.048	0.073	0.028	0.060
DB *IR*	0.031	0.050	0.016	0.033
DR *IR*	0.026	0.025	0.014	0.012
Sl *IR*	0.022	0.035	0.013	0.030
HL *IR*	0.019	0.032	0.011	0.023
Total *IR*	0.452	0.589	0.329	0.489

Table S4. Description, data sources, spatial characteristics and usage rationale of explanatory and predictor variables used in the Random Forest models.

Variable	Code	Type	Resolution	Data source	Description
Forest expansion	FE	Binary	10 m	LC map	Transition to forest occurrence
Elevation	El	Topographic	10 m	DEM tinitaly	Gradient of site suitability
Slope	Sl	Topographic	10 m	DEM tinitaly	Proxy of human pressure
Heat Load Index	HL	Topographic	10 m	DEM tinitaly	Gradient of site suitability
Precipitation	Pr	Climatic	1 km	CHELSA	Average annual precipitation
Temperature	Te	Climatic	1 km	CHELSA	Average annual temperature
Distance to Forests	DF	LU Legacy	10 m	LU 1954	Distance to former forest borders
Distance to Roads	DR	Anthropic	10 m	OSM	Euclidean distance from roads
Distance to Buildings	DB	Anthropic	10 m	OSM	Euclidean distance from buildings
Moving cost	MC	Anthropic	10 m	DEM - OSM	Cost of movement across the terrain

Table S5. Land cover surface as a percentage of the total surface of AL and AP landscapes (10091 and 12830 ha, respectively) divided by regional (CORINE land cover) and landscape (aerial imagery) scales. For comparison purposes, the considered study area is included within the borders of the 16 selected landscapes.

Regional scale (CORINE LC)				Landscape (Aerial imagery)			
AL	1990	2018	Delta	AL	1954	2012	Delta
FO	51.58	53.02	1.44	FO	38.75	48.82	10.07
GR	16.16	4.43	-11.73	GR	28.35	24.51	-3.84
CR	0.81	0.55	-0.26	CR	0.37	0.10	-0.26
UR	0.04	0.04	0.00	UR	0.15	0.30	0.15
UV	31.40	41.96	10.55	UV	32.38	26.26	-6.11
AP	1990	2018	Delta	AP	1954	2012	Delta
FO	67.16	65.98	-1.18	FO	46.99	65.40	18.41
GR	16.03	10.82	-5.21	GR	36.11	26.67	-9.44
CR	3.52	3.66	0.14	CR	11.17	0.98	-10.19
UR	0.65	0.70	0.05	UR	0.95	1.47	0.52
UV	12.63	18.84	6.21	UV	4.78	5.49	0.70